Wyoming Carbon Storage Taskforce Report on Rangeland Research

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Literature Review

Rangelands occupy approximately half of the world's land area and store greater than 10% of the terrestrial biomass carbon (C) and up to 30% of the global soil organic carbon (SOC) (Schlesinger 1997; Scurlock and Hall 1998). On a global scale, rangelands are estimated to sequester C in soil at a rate of 0.5 Pg C yr^{-1} [500 million metric ton C (MMTC)]. Although soil C sequestration rates are generally low on rangelands in comparison to croplands, increases in terrestrial C in rangelands resulting from management can account for significant C sequestration given the magnitude of this land resource. This would signify that modest changes in rangeland C storage have the potential to modify the global C cycle and influence climate (Schimel et al. 1990; Ojima et al. 1993; Conant et al. 2001). Despite the significance rangelands can play in C sequestration, our understanding remains limited (Schuman et al. 2008).

Potential of rangeland to sequester C

Rangelands are defined as uncultivated land areas that support grazing and browsing of animals. A wide range of ecosystems, such as native, perennial grassland, annual grassland, and tundra, are included under rangelands (Lal 2001). Rangelands cover about half of the world's terrestrial area (47%) and contain more than a third of the above- and belowground C reserves (Allen-Diaz 1996). In the United States, 161 Mha of the area is classified as rangeland, which is primarily in the central prairie region with a precipitation gradient from short grass to tall grass prairie. Due to the vast area, rangelands play an important role in soil C sequestration and hence can mitigate climate change due to increasing atmospheric CO_2 concentrations (Follett 2001).

In rangeland ecosystems, rates of C sequestration often peak during early soil formation and diminish with time, approaching a new equilibrium or steady state level (Schlesinger 1990 1995; Chadwick et al. 1994; Schuman et al. 2008). Schuman et al. (2001) estimated that rangelands sequester C at a rate of 11 MMTC yr^{-1}, which can potentially be increased by 8 MMTC yr^{-1} in re-established grasslands. Schuman et al. (2001) reported that about one-third of the U.S. rangelands (54 Mha) have no serious ecological and/or management threats; therefore, the reserve rangeland soil C pool can be considered relatively stable. Changes in the C pool can be possible due to alterations in species diversity. The remaining two-thirds of the rangelands are recognized as facing some constraints that have deteriorated the soil quality and hence C storage. Effects of improved management or restoration of these rangelands will be slow and gradual.

In addition to conserving existing rangeland C pools, it is also important to develop sustainable management practices to increase or restore C sequestration potentials specific for different rangeland ecosystems. For example, recent studies evaluating the impact of grazing on C storage (Manley et al. 1995; Schuman et al. 1999) revealed that in Wyoming rangeland, proper grazing management practices can increase the soil C content at a rate of 0.3 MT C ha^{-1} yr^{-1} compared to ungrazed mixed-grass rangelands, thus saving an additional 17 MMTC yr^{-1}.

The main considerations in rangeland SOC sequestration are: 1) the aboveground C pool is less than 5% of the total ecosystem C storage and turns over every 1 to 2 years and short-term changes of aboveground biomass alone are not likely to affect the C storage; 2) most SOC is recalcitrant and well protected from natural disturbances and generally resists change; 3) a majority of SOC input is due to the decomposition of belowground biomass (roots); and 4) a large perturbation in the SOC pool occurs during soil erosion and with site specific site disturbances (Follett 2001).

The total area prone to high soil erosion may account for 123 Mha (considering rangeland and pasture together). Restoration management practices that improve biomass production may sequester SOC at the rate of 100 to 200 kg ha^{-1} yr^{-1} (Lal 2001). Annual C emission due to erosion is about 14 to 16 MMTC yr^{-1} and restoration can sequester 26 to 41 MMTC yr^{-1}. Restoration treatments may also reduce the volume of eroded sediments and hence C loss (Lal 2001). Several practices that have been shown to impact C sequestration rates on rangelands are shown in Table 1 and suggest that C sequestration rate extremes are from -2.2 to 9.3 Mg C ha^{-1} yr^{-1} with rates generally less than 1 Mg C ha^{-1} yr^{-1} for most practices.

Impact of grazing on rangeland soil C pool

Impacts of grazing management on soil biogeochemical processes that regulate rangeland C dynamics are not well understood due to heterogeneity in rangeland types. Milchunas and Laurenroth (1993) evaluated 34 data sets from around the world to compare soil C of grazed and protected areas and found that about 40% of these results indicate an increase in soil C due to grazing and about 60% showed a decrease or no response to grazing.

Impacts of grazing on ecosystem processes are influenced by: 1) the extent of the removal of photosynthetic biomass (defoliation), which is determined in part by grazing intensity; 2) treading and trampling; and 3) fecal and urine depositions (Heitschmidt et al. 2004). Extent of defoliation depends on plant morphology, growth stages, and the availability of water and nutrients. Repeated grazing reduces plant growth and productivity, whereas light-to-moderate levels cause suppression of growth with occasional growth enhancement (Briske and Richards 1995). Selective defoliation modifies species composition, which often results in lower productivity and undesirable plant compositions. Trampling and treading compact the soil surface increasing the soil bulk density while hoof action deteriorates soil aggregate stability.

Unfavorable changes in soil physical properties may cause a decline in water infiltration and root growth. The addition of nutrients in the form of feces and urine influences soil biogeochemical processes. Altogether, grazing has the potential to influence rangeland C dynamics by altering plant litter chemistry (Milchunas and Laurenroth 1993; Barger et al. 2004), plant biomass allocation patterns (Binodini et al. 1998), litter production, and the spatial distribution of nutrients (Potvin and Harrison 1984; Day and Delting 1990; Frank and Evans 1997). Depending on the intensity, grazing pressure may slow decomposition rates by decreasing plant litter C:N ratio or, due to decreased standing biomass, may accelerate the decomposition by increasing soil temperature (Welker et al. 2004).

Despite the above-mentioned grazing impact on rangeland C dynamics, literature results show a wide variation ranging from positive (Schuman et al 1999) to negative (Dormar and Willms 1998) and no response (Reeder et al. 1998). Gill (2007) evaluated the influence of 90 years of protection from grazing on C dynamics in subalpine rangeland and reported that livestock

Table 1. Measured and modeled rangeland soil C sequestration (C Seq.) rates.

Grassland type – Location	*Management Practice*	*C Seq. rate ($Mg\ C\ ha^{-1}\ yr^{-1}$)*	*Citation*
U.S. Grasslands	*Potential mitigation gains*		Schuman et al. (2001)
	Poorly managed grasslands (113 Mha)	0.1	
	Conservation Reserve Program grasslands (13 Mha)	0.6	
	Potential avoided loss		
	Well managed grasslands (57 Mha)	0.3	
	Poorly managed grasslands (113 Mha)	0.2	
	Conservation Reserve Program grasslands (13 Mha)	0.3	
Short-grass prairie Colorado	Grazing	0.12	Derner et al. (1997)
		0.07	Reeder & Schuman (2002)
Northern mixed-grass prairie Wyoming		0.30	Schuman et al. (1999)
Northern mixed-grass prairie North Dakota		0.29	Frank (2004)
Primarily temperate grasslands Worldwide	Fertilization	0.30	Conant et al. (2001)
	Improved grazing	0.35	
	Conversion from arable to permanent grassland	1.01	
	Conversion from native vegetation to grassland	0.35	
	Introduction of legumes	0.75	
	Earthworm introduction	2.35	
	Improved grass species	3.04	
Cool temperate grassland - USA	Conversion from arable land to permanent pasture	0.27	Post & Kwon (2000)
Managed grassland (simple statistical model) France	Reduction in N-fertilizer inputs in intensive leys	0.3	Soussana et al. (2006)
	Conversion of arable land to grass/legume	0.3 - 0.5	
	Intensification of permanent grassland	0.2	
	Intensification of nutrient-poor grassland	-0.9 to -1.1	
	Permanent grassland to medium-duration leys	-0.2	
	Increasing duration of leys	0.2 - 0.5	
	Short-duration leys to permanent grassland	0.3 - 0.4	
Perennial grassland – Texas	Converted from arable to grassland for 6-60 yr	0.45	Potter et al. (1999)
Prairie grazing lands (11.5 Mha) Canada	Improved management practices (grazing, reduced stocking intensity and N fertilization)	0.04	Lynch et al. 2005
Grasslands - Argentina	Impact of 370 years of livestock grazing (from early European colonization to present)	-2.2	Piňeiro et al. (2006)
Northern mixed-grass prairie South Dakota	Legume interseeding	0.33 - 1.56	Mortenson et al. (2004)
Tallgrass Prairie – Kansas	Nitrogen fertilization	1.6	Rice (2000)
Conservation Reserve Program Wyoming		0.41 - 1.16	Reeder et al. (1998)
Conservation Reserve Program Saskatchewan		5.4 - 9.3	Nyborg et al. (1994)
Conservation Reserve Program Texas, Kansas, Nebraska	Restoration	0.8 - 1.1	Gebhart et al. (1994)
Oklahoma/ Southern mixed-grass prairie		65% ↓ in SOC in 0-10 cm with heavy grazing	Fuhlendorf et al. (2002)
Sudan/ Southern mixed-grass prairie		80% of native rangeland in 100 yr	Olsson & Ardö (2002)
		$MMTC\ yr^{-1}$	
U.S. grazing land	Grazed grassland	29.5 to 110	Follett et al. (2001)
	Land conversion and restoration	17.6 to 45.7	
	Low-input grassland	-4.1 to 13.9	
	'Improved management' and intensification	16.0 to 50.4	
	Emissions from grazing lands	-12 to -19.5	
	Net gain for US total grazing lands	17.5 to 90.5	
Worldwide grassland and Savanna (2,400 Mha)	Present C sink	500	Scurlock & Hall (1998)

grazing had no significant impacts on total soil C or particulate organic matter but active soil C content increased. The loss of C from the active C pool was higher in grazed plots (4.6% of total C) than in ungrazed plots (3.3% of total C). These results imply that grazing may convert the relatively recalcitrant C pool into easily mineralizable C fraction. The exclusion of grazing caused an increase in annual forbs and grasses lacking in dense fibrous rooting system conducive to soil organic matter formation and accumulation (Reeder and Schuman 2002).

The intensity of grazing is a major factor in controlling rangeland SOC dynamics. In pastures of Virginia, USA, Conant et al. (2003) found that soil organic C averaged 8.4 Mg C ha^{-1} more under intensive management or short rotation grazing than extensively grazed or hayed sites. Naeth et al. (1991) observed a negative impact on soil organic matter with heavy intensity or early season grazing as compared to light intensity or late season grazing in the grasslands of Alberta, Canada. Heavy grazing resulted in significant reductions in height of standing and fallen litter, and a decrease in live vegetative cover and organic matter mass. They also reported that large particle size organic matter was associated with ungrazed treatments, whereas, medium or small particle sized organic matter occurred in grazed treatments. Johnston et al. (1970) reported that heavy grazing of fescue grassland range in Alberta changed the color of the A horizon from black to dark brown, reduced the percent organic matter, increased soil temperature but decreased percent soil moisture. Change in organic matter content due to heavy grazing is related to the change in soil physical environment and quantity of organic matter input.

Some studies reported a similar SOC content or no response to heavy grazing. Frank et al. (1995) reported that heavy grazing did not reduce SOC compared with exclosures or control sites, but a moderately grazed pasture contained 17% less SOC than the control to a soil depth of 106.7 cm in the mixed prairie of North Dakota. Preservation of SOC in heavy grazed sites was equal to the exclosures, which was likely due to an increase in blue grama, a species with a dense shallow root system. Heavilygrazed land can be characterized by increased bare ground or dominated by warm-season grasses, forbs, and lichens, whereas light grazing has greater litter cover, and western wheatgrass and other cool-season graminoids composition and production. Manley et al. (1997) found that heavy grazing increased the percent of above-ground biomass contributed by forbs and decreased western wheat grass contribution.

Moreover, Smoliak et al. (1972) observed a higher value in total C and other SOC fractions with heavy grazing compared to light or no grazing. They attributed the outcomes to alteration in the amount of and kind of roots due to changes in species composition and the increased amount of manure deposited by sheep. High intensity grazing replaced the deeper rooted species with moreshallow-rooted ones in thisarid environment. Reeder and Schuman (2002) reported significantly higher SOC in grazed pastures compared to non-grazed exclosures and in the case of short grass steppe, higher SOC was only observed within heavy intensity grazing in semiarid grasslands. Grazing at light to moderate stocking rates resulted in stable, diverse plant communities dominated by forage grasses with dense, fibrous rooting systems favorable for building up SOC. Grazing management practices that encourage forage production also have the potential to increase soil organic matter (Conant et al. 2001).

Consequences of increasing atmospheric CO_2 concentration can affect rangeland C storage by accelerating the photosynthesis rate, hence, increase biomass production, and lowering the decomposition rate (Schuman et al. 2008). An 8 year old CO_2 enrichment study in tall grass prairie revealed that the soil system sequestered an additional 59 g C m^{-2} yr^{-1} due to increased above and belowground production and resulted in higher SOC. The extent of increased CO_2 sequestration potential of rangeland depends on the response of plant communities to elevated CO_2. Grass species having higher leaf area than their competitor, will become increasingly dominant under elevated CO_2 concentration (Teyssonnneyre et al. 2002). Dominance of C_3 species over C_4 species is another probable consequence of increased CO_2 concentration (Soussana and Lüscher 2007). Warm-season (C_4) grasses are less nutritious than C_3 grasses in terms of crude protein content and have a higher C:N ratio. Changes of grazer food quality in terms of fine scale (crude protein concentration and C:N ratio) and coarse scale (C_3 species vs. C_4 species) may be expected due to elevated CO_2 concentrations. The three main issues in considering the fate of rangeland in a high atmospheric CO_2 concentration, are 1) changes in production and quality of herbage; 2) changes in the global environment such as rising temperatures, changing precipitation and rising CO_2 concentrations will become determinant factors in plant community diversity and loss of production; and 3) the impact of extreme climatic conditions e.g. heat waves and droughts on net C exchange in terrestrial ecosystem (Ciais et al. 2005; Soussana and Lüscher 2007). The preservation of rangeland C storage in an altered climate with a high temporal variability and elevated CO_2 concentrations, which may saturate SOC sinks, will be the most sensitive issue in the future.

Rangeland Research Activities

Northern Mixed-Grass Prairie, Cheyenne, WY

To develop a better understanding of the influences of management and climate on terrestrial C sequestration in rangelands in the Great Plains, research was undertaken on selected sites at the High Plains Grasslands Research Station (HPGRS) for the Big Sky C Sequestration project. Schuman et al. (1999) initiated research on a northern mixed grass rangeland site in 1993 and found that grazing significantly increased SOC in the upper 30 cm of pastures grazed season-long at light- and heavy-stocking rates, compared to non-grazed exclosures. Sampling protocol, experimental design, and laboratory methods are described by Schuman et al. (1999). They estimated that over 11 years of grazing, these pastures sequestered C at the rate of 0.30 Mg C $ha^{-1}yr^{-1}$. This research was continued under this project and sampling was carried out in 2003 to assess the effect of several drought years during the period 1993-2002. Re-sampling of the permanent transects established by Hart et al. (1988) in 1983 and sampled in 1993 showed that drought can significantly impact rangeland SOC levels; hence, C sequestration. SOC in the 0-30 cm soil depth of the continuous, heavily grazed (CH) and non-grazed (EX) was significantly lower than that of the continuous, lightly grazed (CL) treatment in 2003.

SOC levels present in the various soil depths in 1993 and 2003 of the northern mixed grass rangeland site near Cheyenne, Wyoming are shown in Table 2. Significant below average precipitation in 7 of between those 10 years (Figure 1) resulted in a loss of SOC from the CH and EX treatments. During the 21 years of grazing on these pastures plant community composition changes and productivity occurred that help to explain the loss in SOC observed during these drought years. In the past 21 years plant productivity in the heavily grazed pastures has declined by nearly 50% compared to the EX and CL but even more important is the fact that the CH has become dominated by C_4 grasses (predominately blue grama) compared to the other two treatments that are dominated by C_3 grasses (western wheatgrass).

Table 2. Soil organic carbon mass in the non-grazed (EX), continuous, lightly grazed (CL), and continuous, heavily grazed (CH) pastures at the High Plains Grasslands Research Station, Cheyenne, WY in 1993 and 2003, values in ( ) are percentage change from 2003 to 2006 (Modified from Ingram et al. 2008).

Soil depth	1993†			2003		
(cm)	EX	CL	CH	EX	CL	CH
	(kg ha^{-1})					
SOC						
0-15	28,163^{b}	35,141^{a}	35,950^{aA}	27,296^{a} (–3.1)	31,961^{b} (-9.0)	25,971^{aB} (-27.8)
0-30	47,924^{b}	57,988^{a}	58,298^{aA}	47,293^{a} (-1.3)	54,193^{b} (-6.5)	42,521^{aB} (-27.1)
0-60	88,147^{b}	91,936^{b}	101,268^{aA}	80,456^{a} (-8.7)	92,471^{b} (+0.6)	70,526^{aB} (-30.4)

Means within a soil depth and year with different lowercase letters are significantly different at $P \leq 0.10$.
Means with a soil depth across years with different uppercase letters significantly different at $P \leq 0.10$.

A significant reduction in aboveground biomass in the CH resulted in lower potential C inputs from both above- and belowground litter production (Schuman et al. 1999). Potential root C inputs also changed due to shift in plant community composition. About 56% of western wheatgrass root biomass occurs in the top 15 cm and 86% in the top 60 cm of the soil profile (Weaver and Darland 1949). Needleandthread (C_3) roots are also predominately found in the upper 45 to 90 cm (Coupland and Johnson 1965). However, 83% of blue grama roots are found in the surface 15 cm of the soil with approximately 92% of their roots being in the surface 30 cm (Weaver and Darland 1949). The fact that the abundance of blue grama roots are in the surface 30 cm of the soil allows this species to take advantage of the small precipitation events that are common in this environment and they deposit a greater amount of their C belowground (Coupland and Johnson 1965). With blue grama the predominance of shallow roots make them more vulnerable to decomposition and oxidation and eventual loss to the atmosphere as CO_2 and CH_4. Bare ground in the CH treatment also increased by 50-90% relative to the CL and EX treatments and resulted in the potential loss of C from the soil via wind and water erosion (Neff 2005).

Changes in the quantity, quality and location of the C seem to be important factors in regulating C pools in grazed ecosystems (Ingram et al. 2008). These findings are supported by recent research that shows that climate change (temperature and precipitation) can be responsible for significant SOC losses (Bellamy et al. 2005). Soil temperature data collected from an ungrazed area near the study site showed a general increase in temperature at both the 38 and 102 mm soil depth (data not shown) during the study period 1982-2003. The flux of C from the soil to the atmosphere during periods of low precipitation and/or drought in comparable ecosystems has been documented by others (Meyers 2001; Frank 2004; Hunt et al. 2004; Morgan et al. 2004; Svejcar et al. 2008).

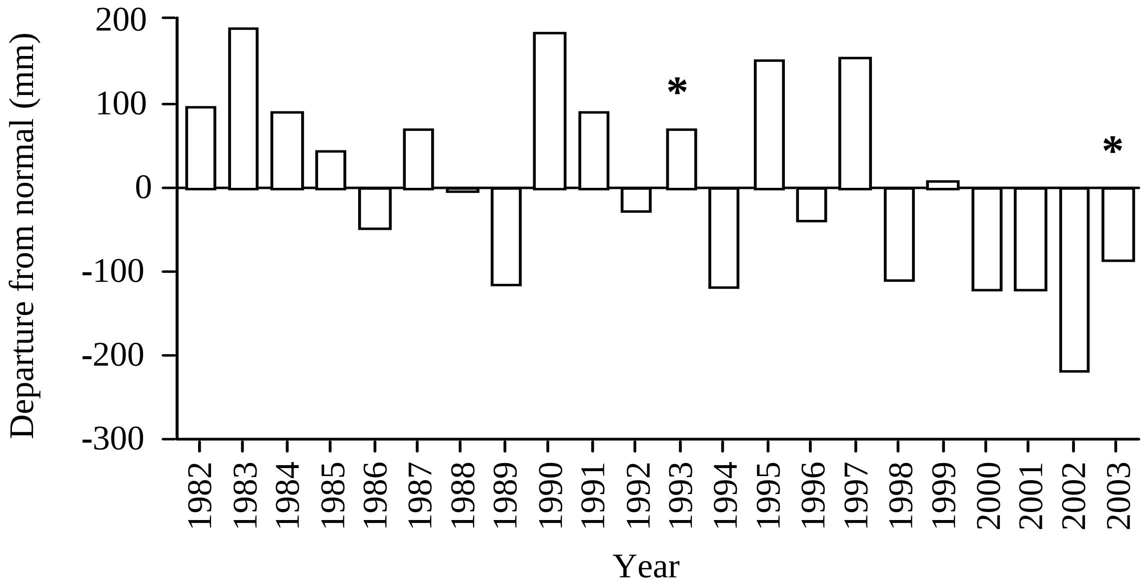


Fig. 1. Departure from the long-term (1982-2003) mean annual precipitation (381 mm)at the HPGRS, Cheyenne, WY. The study was initiated in 1982 and sampled in 1993 and 2003 (indicated by the asterisks) (Ingram et al. 2008).

To further evaluate the long-term effects of livestock grazing and short-term average precipitation effects on SOC we re-sampled the northern mixed grass rangeland site at the HPGRS in the spring of 2006 (Schuman et al. 2007). Precipitation in 2003 and 2004 were slightly below the long-term average and 2005 precipitation was above the long-term average. SOC in the 30 cm soil surface showed no differences between treatments in 2006. The CL and CH grazing treatments showed a slight but significant decrease in SOC in 2006 compared to 2003; while the EX showed no change during that time. We believe that during the time period 2003 to 2006 the climatic conditions may have overridden management effects on SOC due to the ecological lag of the severe drought of 2002. Conical multivariate analysis of the microbial community data indicates that the structure of the microbial communities were statistically different among the grazing treatments (Figure 2). If we assume the CL grazing treatment is more typical of the grazing that occurred over the millennia that these rangelands experienced prior to settlement, the EX and CH appear to have shifted away from the CL treatment. The microbial biomass, microbial respiration and N-mineralization rates also responded similarly, CL>EX>CH (Ingram et al. 2008).

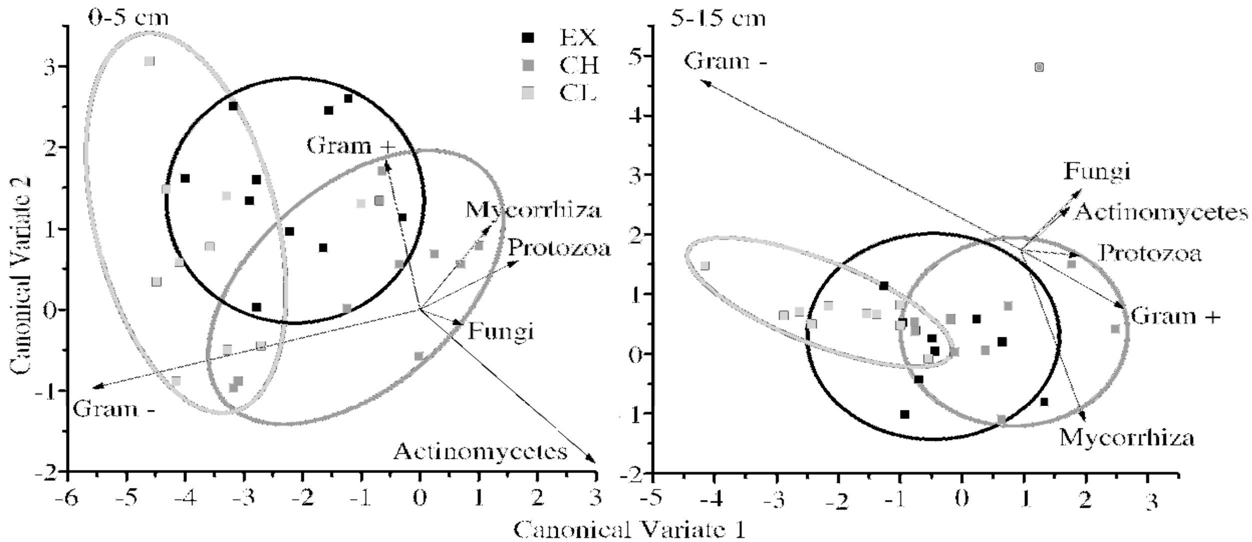


Figure 2. Canonical multivariate analysis for microbial groups as determined by phospolipid fatty acid analysis (PFLA) for the 0-5 and 5-15 cm depth increments from soils in non-grazed (EX), continuously, lightly grazed (CL), and continuously, heavily grazed (CH) pastures at the High Plains Grasslands Research Station, Cheyenne, WY. Ellipses were hand drawn to indicate grazing treatment groupings (Schuman et al. 2007).

Evaluation of HPGRS SOC dynamics in CL and CH grazed sites and the 60 yr old EX (Ganjegunte et al. 2005) found that within surface soils (0-5 cm) the SOC and N contents were significantly greater in CL (SOC -2.57% and soil total nitrogen (SN) - 0.23%) than CH (SOC – 1.98% and SN - 0.17%) or EX (SOC - 2.09% and SN - 0.18%). Significant lignin (cupric oxide (CuO) analysis) contents (e.g., Vanillyl + Syringyl + Cinnamyl compounds - VSC) were noted in EX (429 mg kg^{-1} soil) than CL (314 mg kg^{-1} soil) and CH (294 mg kg^{-1} soil) soils. The CuO oxidation of humic (HA) and fulvic (FA) acids indicated that HA under CL contained significantly greater V, S, and total lignin than that under CH or EX, whereas FA extracted from CH contained significantly greater V and C than that extracted from CL and EX. The ^{13}C NMR spectra of HA did not vary significantly among the three grazing treatments nor did the FA spectra. However, overall the HA spectra had significantly greater alkyl, methoxyl, and aromatic C than FA and FA spectra had significantly greater O-alkyl and di-O-alkyl C than HA, suggesting HA is more recalcitrant and aromatic than FA. The δ^{13}C and δ^{15}N values for humic or fulvic acid did not differ significantly among grazing treatments; however, the overall δ^{15}N value for HA (+2.9) was significantly lower than that for FA (+4.6) indicating N in HA is not readily available to plants. These results suggest that there are beneficial effects of light grazing compared to heavy grazing and non-grazing with respect to increased SOC and N contents. Stimulation of aboveground vegetation growth, better incorporation of aboveground biomass, and increased decomposition rates of plant residues occurred within the CL grazing and increased SOC contents.

To further assess the effects of climatic and land management effects on soil C sequestration, Derner and Schuman (2007) reviewed the literature to correlate C sequestration with precipitation and management practices. No statistical relationship was found between length of grazing practice and change in SOC. The general trend suggested a decrease in C sequestration with longevity of the grazing practice across stocking rates (Figure 3). This trend is consistent with the understanding that the ecosystem will reach a 'steady state' and changes in inputs or management would be required to sequester additional C (Conant et al. 2001, 2003; Swift 2001). Derner and Schuman (2007) suggest that C sequestration would stop after 80-85 years with a grazing practice. Mortenson et al. (2004) suggested that a 'steady state' was reached after about 30-35 years after interseeding a legume into native rangelands. The effects of precipitation gradient on rangeland C storage in the 0-30 cm soil depth showed a general increase with increasing precipitation (Derner and Schuman 2007). Comparisons of grazed vs nongrazed pastures revealed a threshold from positive to negative sequestration occurs at 440 mm when assessing the 0-10 cm soil depth, and at 600 mm precipitation when assessing the 0-30 soil depth (Figure 4). Above these precipitation levels it appears that C sequestration in rangelands decreases. These evaluations of precipitation effects on C sequestration agree with observations of Sims et al. (1978) and Sala et al. (1988) in identifying the 370 to 400 mm precipitation range as a transition where aboveground ecosystem responses to grazing occurs.

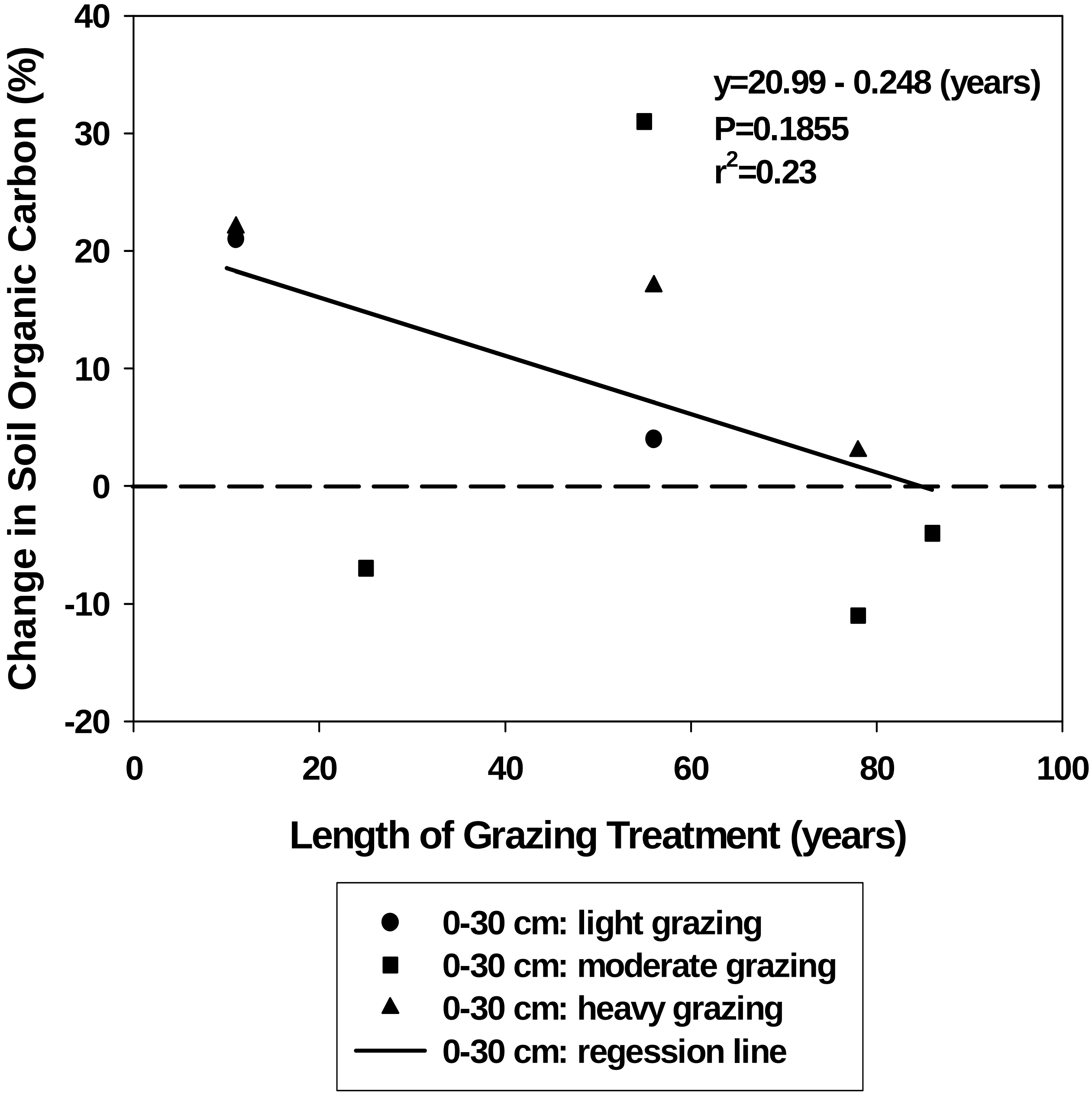


Figure 3. Change (grazed vs. non-grazed, %) in soil organic carbon with respect to length of grazing treatment in North American Great Plains (data is from Frank et al. 1995; Schuman et al. 1999; Reeder and Schuman 2002; Derner et al. 2006) (From Derner and Schuman 2007).

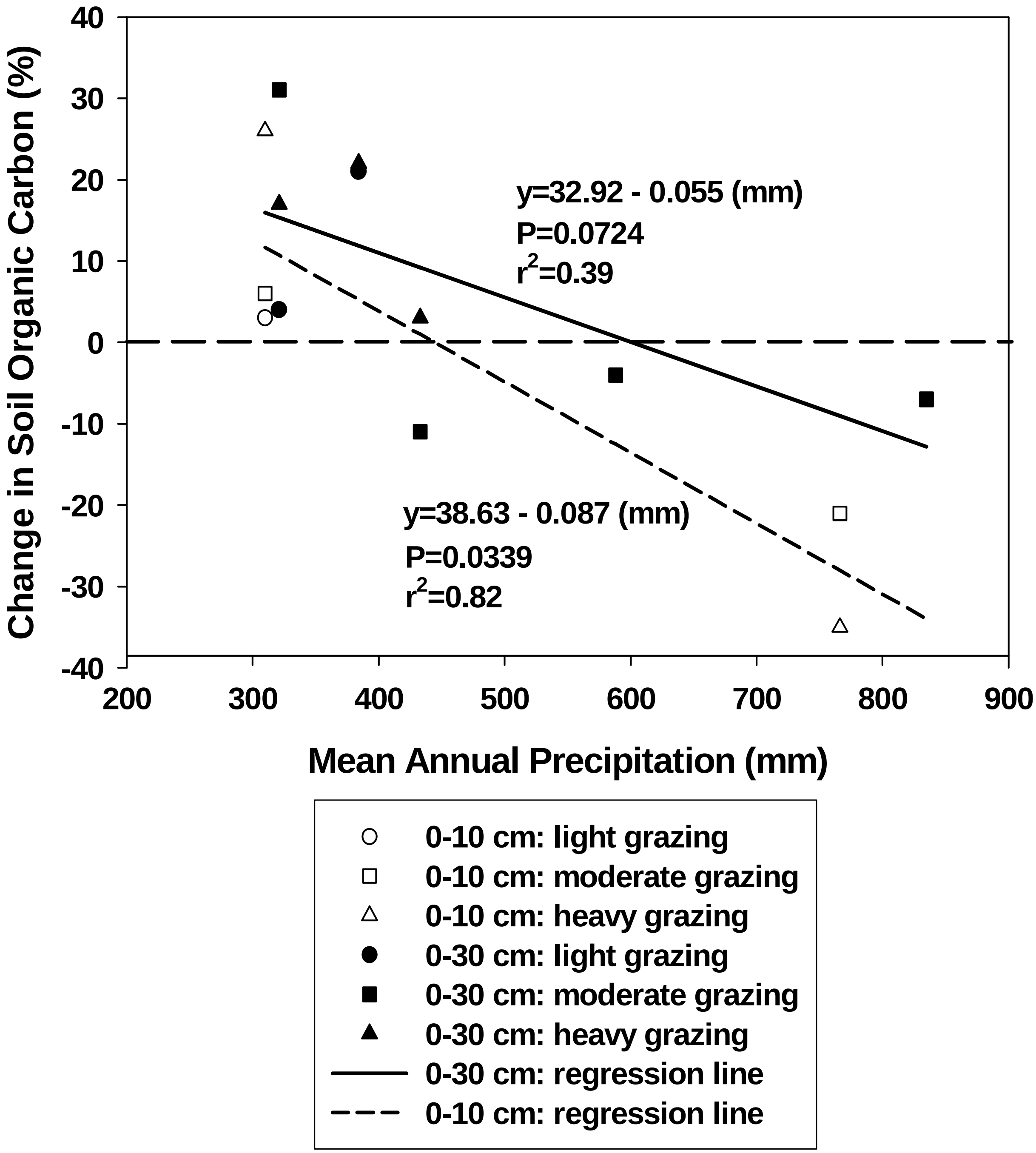


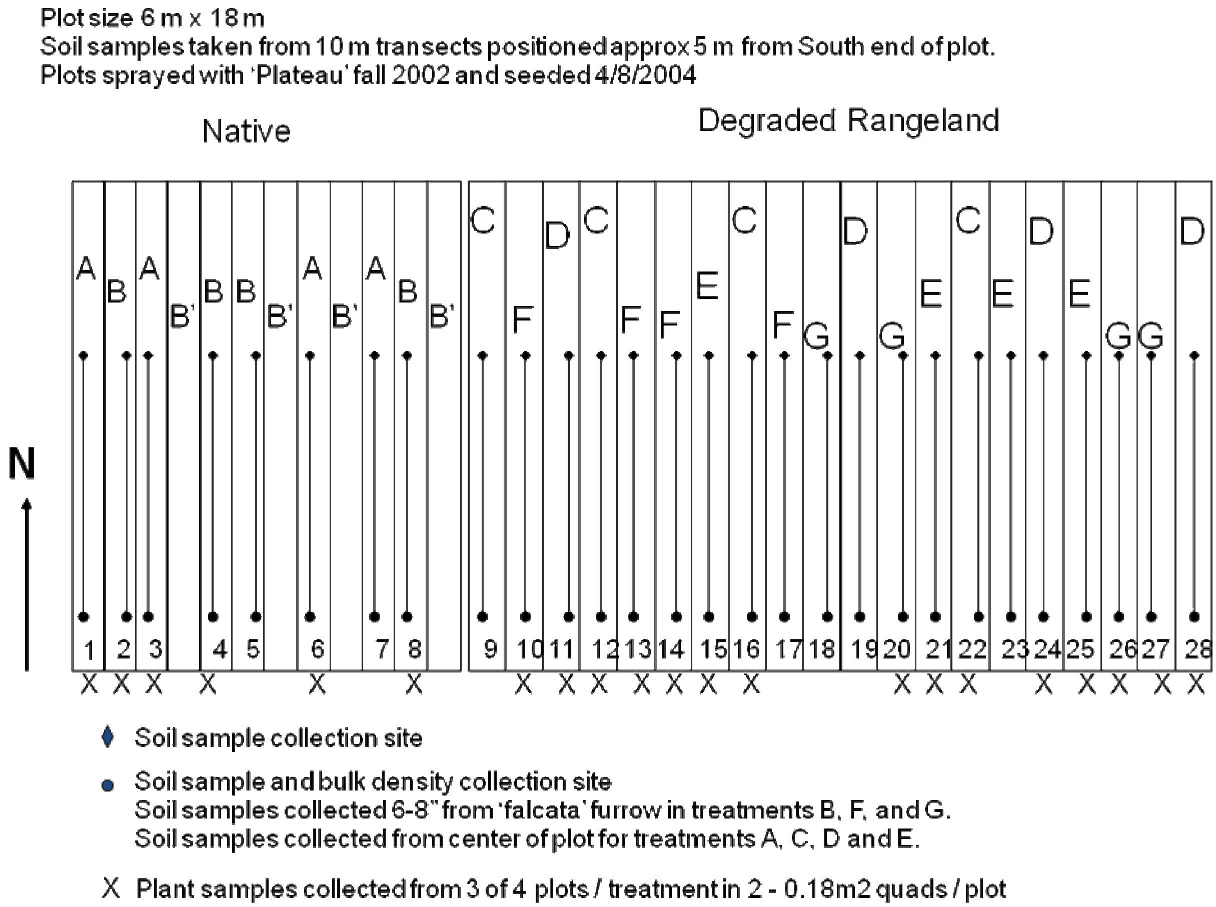
Figure 4. Change (grazed vs. non-grazed, %) in soil organic carbon change with mean annual precipitation (MAP) with grazing in North American Great Plains (0-10 cm data is from Smoliak et al. 1972 and Fuhlendorf et al. 2002; 0-30 cm data is from Frank et al. 1995; Schuman et al. 1999; Reeder and Schuman 2002; Derner et al. 2006) (From Derner and Schuman 2007).

Browder Ranch Study, Lusk, WY

A study was initiated at the Terry Browder Ranch in 2003 to assess the potential of interseeding yellow blossom alfalfa (*Medicago sativa* ssp. *falcata)* into native rangeland and go-back lands (lands disturbed by farming that have naturally returned to a grassland). The go-back lands had severe infestations of downy brome grass *(Bromus tectorum)*. Treatments included herbicide application to control the downy brome, interseeding of alfalfa, interseeding of native grass mixture, interseeding of native grass mixture and alfalfa and no treatment (control) plots (Figure 5). The herbicide was applied in September 2002 and the interseeding was accomplished in the spring 2003. Unfortunately, 2003 and 2004 were extremely dry years and very little alfalfa became established in any of the plots. However, we did see a good response in forage production where the herbicide was applied to control the downy brome grass (Table 3). Forage production was monitored for several years after treatments were imposed but were discontinued after 2006 because of a lack of differences among treatments.

Sagebrush-Grassland Carbon Evaluation-Kemmerer, WY

Much of the research literature dealing with C sequestration on rangelands has included several grassland ecosystems but not shrublands or shrub-grasslands. Because of this lack of data and published literature the Chicago Climate Exchange would not include these lands in their recently published Rangeland Offset Program for trading C offset credits. To assist in filling this gap in the knowledge base relating to rangeland C sequestration. Drs Justin Derner and Gerald Schuman, in cooperation with BLM and NRCS have located several sites in western Wyoming near Kemmerer to sample in the spring of 2009. These include sagebrush-grassland sites that have been mowed at various heights or treated with chemicals to control sagebrush growth. One of the sites includes two levels of grazing using different grazing management and stocking rates. It is their intention to sample these sites using the same experimental methods and designs used at the northern mixed-grass prairie site near Cheyenne. Some funds are available to carry out this effort, but additional funds are being sought to expand the sampling and sample analyses.



<u>Treatments:</u>

<u>Native rangeland</u>

A - Control
B - Interseeded falcata
B'- Furrowed, no falcate interseeded

Disturbed rangeland (go-back)

C - Control
D - Herbicide only
E - Herbicide + grass seeded
F - Herbicide + falcate seeded
G - Herbicide + grass and falcata seeded

Figure 5. Experimental design of the research plots at the Terry Browder Ranch, Lusk, WY

Table 3. Plant production as affected by the various rangeland treatments on the Terry Browder Ranch, Lusk, WY.

	Treatment biomass means (kg/ha)						
Plant group	A	B	C	D	E	F	G
2004							
Cool-season grasses	397	341	579	860	888	944	1231
Warm-season grasses	125	175	182	92	92	216	131
Annual grasses	83	91	97	59	7.1	26	14
Perennial forbs	96	22	17	81	14	94	54
Annual forbs	1.3	3.6	0.1		0.8	7.0	
Fringed sage			156		12	100	0.9
Standing dead	237	121	757	290	548	767	481
Litter	577	736	1632	1535	1426	2412	1433
2005							
Cool-season grasses	1147	975	2228	2455	3577	2499	3378
Warm-season grasses	214	256	56.5	149	202	158	165
Annual grasses	150	229	322	487	142	584	275
Perennial forbs	43	166	210	156			127
Annual forbs	328	195	2.6	84	22	7.5	0.8
Fringed sage			67	206	19	405	178
Standing dead	13	13	119	46	81	173	14
Litter	630	423	1020	1688	1019	1365	1284
2006							
Cool-season grasses	933		629	721	1382		
Warm-season grasses	361		399	151	75		
Annual grasses	7.0		123	169	49		
Perennial forbs	18		21	58	1.7		
Annual forbs	0.2		0.7	0.1	0.1		
Fringed sage	50		181	109	88		
Standing dead	222		606	630	1103		
Litter	631		2099	2850	1873		

Carbon Credit Marketing

Recent efforts by a scientific advisory panel to the Chicago Climate Exchange (http://www.chicagoclimatex.com/) resulted in a Rangeland Soil Carbon Offset Program. Dr.'s Justin D. Derner and Gerald E. Schuman were invited to serve on that advisory panel which developed the protocol being used by the Chicago Climate Exchange to buy C offset credits on rangelands in the Great Plains and Northwestern rangelands. Their participation greatly enhances the efforts of this overall research project. The map below (Figure 6) shows the Land Resource Regions for which these protocol were developed and the relative C sequestration rates accepted by the Chicago Climate Exchange. For greater detail on this subject see the above mentioned web site. Sagebrush-grasslands were not included in this overall effort because scientific data is not available to determine or establish C sequestration rates on those ecosystems. This is an area of research in this specific rangeland ecosystem that merits attention.

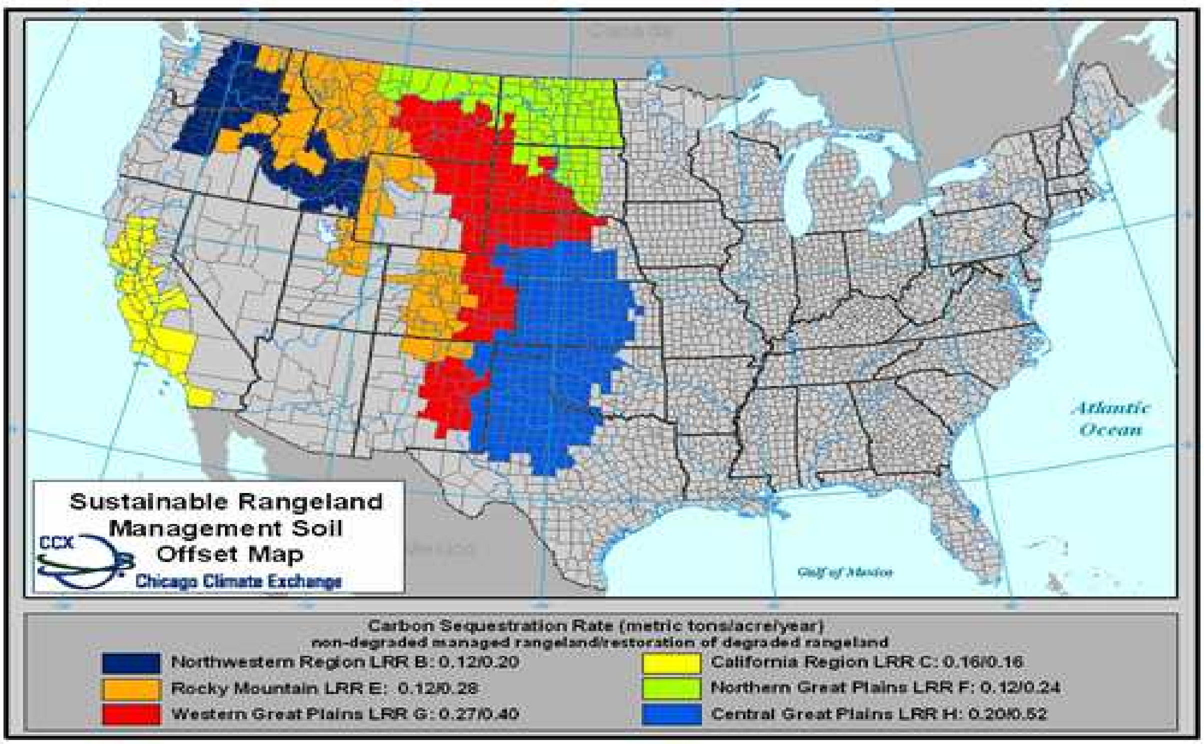


Figure 6. Land Resource Regions and their relative C sequestration rates according to the Chicago Climate Exchange.

Examples of Management Practices for C Sequestration – Western Great Plains

A rancher decides to implement a prescribed grazing Best Management Practice (BMP) on an area of rangeland in the western Great Plains. Proper stocking rates are applied, animal distribution is enhanced and seasons of use are rotated among years to foster increased health and sustainability of the rangeland. The amount of carbon sequestered averages approximately

0.27 Mg C ha^{-1} yr^{-1} (Figure 6). If the rancher in this same Land Resource Region (western Great Plains) had prior degraded rangeland and implemented BMP for restoration of these lands, the amount of carbon sequestered averages approximately 0.40 Mg C ha^{-1} yr^{-1}. Using the range of carbon trade values from the Chicago Climate Exchange web site this year (2008), values of sequestered carbon are given for both non-degraded managed rangeland and restored degraded rangeland in the western Great Plains (Table 4).

Table 4. Values of sequestered carbon for rangelands in western Great Plains ($/acre),

	Values of Sequestered Carbon for rangelands in western Great Plains ($/acre)						
	Dollars per metric ton CO2						
	$1/ton	$2/ton	$3/ton	$4/ton	$5/ton	$6/ton	$7/ton
Rangeland Category							
Non-degraded managed	0.27	0.54	0.81	1.08	1.35	1.62	1.89
Restoration of degraded	0.40	0.80	1.20	1.60	2.00	2.40	2.80

Other potential management practices that could result in an increased C sequestration in plants and soils are listed in Table 5.

Table 6. Practices that result in a net sequestering of carbon in plants and soils	
GRAZING LAND PRACTICES	Accumulation Mg ha^{-1} yr^{-1}
Controlling annual plants	+
Seeding areas of low vegetation density with perennial plants	+
Improving/maintaining range health to a high level by proper stocking rates[1]	0.30
Improving/maintaining range health to a high level by prescribed grazing [1]	>0.30
Rotation grazing	+
Facilitating grazing management by developing livestock water facilities and fencing	+
Interseeding with legumes[2]	+
Reduce the amount of brush on shrub/tree-dominated areas, preferably by mechanical methods	+
Maintain healthy grazing lands by implementing practices suggested by results of periodic vegetation condition assessments	+
Planting tree species that produce highest amounts of wood	+
Selecting sites for new plantings according to the highest potential for growth rates	+
Plant shrubs and trees when reclaiming disturbed areas, such as mine lands	+

Farmstead shelterbelts[3]	14 MMTC in 20 years
Living snow fences[3]	0.192 MMTC per 1600 km in 20 years
+ Carbon Storage is increased, but research is needed to determine values	

[1(]Eve et al. (2002); [2(]Mortenson et al. (2004); [3]/National Agroforestry Center (2001)

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